

## InLCA: Selected Papers

# Comparison of Two Equivalency Factor Approaches with Simplified Risk Assessment for LCIA of Toxicity Impact Potential

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**Abstract.** Three approaches recommended for characterization of toxicity impact potential in a life cycle impact assessment (LCIA) are tested on a case study and compared. The two equivalency factor methods are the Persistence, Bioaccumulation, and Toxicity (PBT) method and the Multimedia Fate Modeling (MFM) method using a Mackay Level III model with state-specific environmental data. The simplified risk assessment (SRA) method involved dispersion modeling using site-specific environmental data. The life cycle inventory information evaluated by all three methods was limited to manufacturing of the RDX-based explosive in Kingsport, Tennessee. The effort to collect site-specific environmental data and conduct air dispersion modeling for the SRA method required about 24 times more effort than the PBT method and about 4 times more effort than the MFM method. Direct comparison of impact potential scores for the three approaches were limited to inhalation toxicity scores for nine air pollutants modeled by SRA. Correlations were made on the rank order of the impact potential scores for the nine air emissions evaluated for all three LCIA methods. Although the number of chemicals compared is very limited, the best correlation coefficient (0.96) was between the rank orders for the MFM and the SRA methods. The minimal effort and reduced accuracy of the PBT approach make it best suited for screening large numbers of chemicals for further evaluation of the highest ranked chemicals. The intermediate effort and reasonable accuracy (includes transfers to other media) of the MFM approach make it well suited for LCIA's involving comparative assertions or governmental policy decisions. The maximum effort and assumption of highest accuracy make the SRA approach suitable only after limiting the locations of interest to a few sites by screening with the other two approaches.

**Keywords:** Air dispersion modeling; aquatic toxicity; equivalency factor; fugacity; InLCA; inhalation toxicity; Life Cycle Impact Assessment (LCIA); LCIA; multimedia fate model; simplified risk assessment; terrestrial toxicity; toxicity impact potential

## 1 Introduction

### 1.1 Characterization for life cycle impact assessment

A life cycle impact assessment (LCIA) has been defined by the Society of Environmental Toxicology and Chemistry (SETAC 1993a, 1993b, and 1997) and the International Organization of Standardization (ISO 1999) as a relative approach based on a functional unit for characterizing and

assessing the potential effects of the environmental burdens (resources used and emissions released) identified in a life cycle inventory (LCI). As modified by ISO (1999), an LCIA consists of three mandatory elements (scoping including impact category and model selection, classification, and characterization), which can be followed by four optional elements (normalization, grouping, weighting, and data quality analysis). This paper deals with the characterization element of LCIA, which includes modeling, quantification, and aggregation of inventory data within a particular impact category.

Equivalency factors (EFs), which are called characterization factors by ISO, are frequently used as part of the characterization step to estimate the contribution of specific inventory data to the impact categories to which they have been assigned during classification. Within this context, EFs have been defined as factors derived from a model which convert assigned LCI results to the common unit of the category indicator and allow them to be aggregated into a single impact category (SETAC 1997, ISO 1999). Generally, EFs are intended to quantify the relative severity of a specific inventory item's contribution to an impact.

Recommendations to include regional scaling (Owens 1996, Hogan et. al. 1996, Udo de Haes 1996) as part of the characterization step have been identified. Regional scaling factors incorporate consideration of ecosystem characteristics (temperature, soil type, etc.) into the characterization of LCI data. Tolle (1997) has proposed and applied a methodology for incorporating regional scaling in LCIA for five different impact categories. This work further refines and evaluates the methods of Tolle (1997) as related to toxicity associated impacts.

### 1.2 Three toxicity evaluation approaches for LCIA

This study compares two EF approaches and a simplified risk assessment (SRA) approach for characterizing toxicity-associated impacts. The two equivalency approaches are supported by algorithms that incorporate toxicity benchmarks (e.g., lethal concentrations and doses) and data on fate/potential exposure. Each of the three approaches are based on one of the methods suggested in LCIA framework (SETAC 1993a) and state-of-the-art (SETAC 1997) documents. The first approach, the Persistence, Bioaccumulation, and Toxicity (PBT) method, systematically incorporates data

on these three parameters into a single EF for each inventory item. The second approach, the Multimedia Fate Model (MFM) method, estimates the steady-state concentration of a chemical inventory item in air, water, soil, sediment, and biota resulting from the continuous release of the chemical to one of these environmental media using Level III MFM concepts (Mackay 1991 and Mackay et al. 1992). The concentrations estimated by MFM are then equated to potential doses to specific receptors. The third approach, the SRA method, involved dispersion modeling using site-specific environmental data to determine environmental concentrations of emissions and compare them with threshold values considered safe for humans and wildlife. The SRA method has been simplified by excluding emissions from dispersion modeling, if the release concentration is already below a threshold concentration considered safe.

While a number of published articles have compared different scoring and modeling approaches for LCIA, not many case studies have been published in which the inventory data and equivalency scores are combined prior to comparison. Examples of other published approaches for assessing toxicity related impacts by combining toxicity benchmarks and MFM results have been developed by researchers at CML (Centre for Environmental Science, Leiden) (Guinée and Heijungs 1993) and implemented by the Environmental Defense Fund (EDF) (Hertwich et al. 1998). EDF's human health risk scoring method<sup>1</sup> uses CalTOX, an integrated environmental fate and exposure model, to estimate the dose an individual could receive after a unit amount of a chemical is released to air or water. The EDF method assesses contribution to non-cancer health effects using toluene as a reference substance. The EDF human toxicity, impact potential method was compared with three less-comprehensive methods by Hertwich et al. (1998) based on calculation of pentachlorobenzene/styrene equivalency potential ratios. Additional studies using a MFM approach have been described by Guinée and Heijungs (1993) and Guinée et al. (1996), respectively, using phenol or 1,4-dichlorobenzene as reference substances.

### 1.3 Selection of case study

The SRA approach to LCIA requires a substantial effort, including detailed environmental characterization data and emission release information for each site evaluated. Thus, it was desirable for this evaluation to select only one site in a life cycle that had relatively easy access to this detailed information and also had a significant number of different chemical emissions for comparing toxicity characterization approaches. Data were already available for an LCI (U.S. EPA 1998a) and LCIA (U.S. EPA 1998b) of the GBU-24 munition, which contains an RDX-based explosive core. Within the life cycle of the GBU-24 munition, the Holston Army Ammunition Plant (HSAAP) in Kingsport, Tennessee had the best combination of these requirements and was selected for the case study.

<sup>1</sup> EDF's human health risk scoring method is described at <http://www.scorecard.org/>

## 2 Methods

### 2.1 Selection of toxicity benchmarks

Within this study, acute toxicity benchmarks were used for both of the EF approaches, while chronic toxicity threshold concentrations were used for the SRA approach. Acute toxicity benchmarks were used for the PBT and MFM approaches to assess three toxicity impact categories previously described by Tolle (1997). The Inhalation Toxicity impact category represents human health toxicity and the Terrestrial and Aquatic Toxicity impact categories represent ecological toxicity.

Acute toxicity values for the most sensitive species (lowest LD<sub>50</sub> or LC<sub>50</sub>) were used to develop EFs for each of the three toxicity impact categories. It is recognized that use of the lowest LD<sub>50</sub> value presents an additional level of uncertainty, since some chemicals have been tested on a greater variety of species than others and are therefore more likely to include toxicity benchmarks on highly sensitive species. However, the toxicity benchmarks used are restricted to only a few commonly tested species, i.e., rodents (typically rats or mice) for Inhalation Toxicity and Terrestrial Toxicity and freshwater fish (fathead minnow preferred and trout excluded) for Aquatic Toxicity. It is also recognized that toxicity benchmarks are limited because they do not consider subtle biological endpoints of long-term, low-level chemical exposures, such as untested transgenerational and endocrine-disrupting effects (Kimerle et al. 1997). Another limitation of toxicity benchmarks is that they are typically based on laboratory studies conducted under controlled environmental conditions with a single parent test chemical, and do not consider degradation products. Therefore, toxicity benchmarks may not reflect potential effects under actual environmental conditions, including the possibility of additive, synergistic, or antagonistic effects from combination with other pollutants or with degradation products.

Although it is recognized that chronic measures of toxicity appear better suited to the assessment of impacts related to the environmental exposures as modeled within the fate/exposure components of both approaches, chronic toxicity values (including estimated QSAR values) are unavailable for many chemicals, particularly for terrestrial species. In fact, of the 2,863 U.S. organic chemicals produced or imported in volumes over 2.2E+6 kg (1E+6 pounds) per year, chronic toxicity benchmarks are available for only 13.9%, while acute toxicity benchmarks are available for 49.4% (Betts 1998). As a result, acute benchmarks were used (1) for the purpose of consistency among benchmarks and (2) as opposed to omitting the chemicals from the assessment. Although using acute-to-chronic conversion ratios or other conversion methods (Kimerle et al. 1997) is not considered, it may be an appropriate addition to future studies where chronic data with identical endpoints are available for many chemicals.

Finally, wildlife and fish, as opposed to human, toxicity benchmarks have been applied. Again, the lack of sufficient data is the primary reason for this approach. In addition, the diet for wild species is typically restricted to a small area in which water supplies are untreated. Furthermore, this approach is considered conservative with respect to human exposure be-

cause areas with heavily polluted soil and water are often restricted from human access and drinking water is monitored and treated in municipal systems. Also, toxicity benchmarks from animal tests are frequently used to set safe food and drinking water chemical concentration levels for humans.

Chronic toxicity threshold concentrations were used for the SRA approach instead of the acute toxicity benchmarks used for the two approaches involving EFs. Chronic toxicity threshold concentrations have the advantage that they are more relevant to the chronic release of emissions over a long time period, which is typical for emissions associated with normal operation of manufacturing facilities. The threshold concentrations [e.g., Federal Water Quality Criteria (WQC), National Ambient Air Quality Standards (NAAQS), reference doses (RfDs), and minimum risk levels (MRLs)] considered in this study included toxicity indices developed for regulatory and SRA purposes, which are not considered appropriate for an LCIA where individual chemical toxicity scores are combined into a category indicator result (Udo de Haes and Jolliet 1999, Owens 1999, ISO 1999). These threshold concentrations typically include safety factors that are value based and are not strictly science based like an  $LD_{50}$ . Also, the endpoints are not uniform. For different chemicals the threshold concentration may be based on endpoints that vary from an allergy to death. Combination of these types of threshold concentrations, which involve different types of toxicity, into a single impact category is inappropriate for LCIA characterization because it involves value choices (ILSI 1996). For this application to a SRA, the threshold concentrations were used despite the disadvantages, because the toxicity evaluation for each chemical was not combined into a single category indicator result.

## 2.2 PBT approach

The PBT approach used in this study for calculating EFs combines toxicity and fate hazard values (HVs) for the three toxicity impact categories described above and slightly modifies the approach described by Tolle (1997). This approach is based on a U.S. EPA (1994) chemical hazard evaluation document prepared by the University of Tennessee and summarized by Swanson et al. (1997). The EF is multiplied by the associated inventory quantity to arrive at a score for each inventory item's contribution to the impact at hand. The scores may be added when different locations or life cycles are being compared or considered.

For the PBT approach, the log of each toxicity benchmark is used to establish the toxicity HV. The toxicity HV is assigned a value of 0.1 or 5, respectively, if the benchmark is above or below specified threshold values. In this study, 0.1 has been applied as the minimum toxicity HV instead of zero, when the toxicity value is below the threshold. This is done to account for the potential for threshold exceedance when combined with background levels of the same chemical or for synergistic effects when combined with emissions of a different chemical. The toxicity HVs for toxicity benchmarks between these threshold values are determined from the formulas indicated in the publications by U.S. EPA (1994) and Swanson et al. (1997). This differs from the MFM and

SRA approaches discussed below, in that an exposure concentration is not calculated and is not divided by a threshold toxicity benchmark.

Fate and potential exposure is incorporated into the PBT assessment using two measures of persistence [biological oxygen demand (BOD) half-life and hydrolysis half-life] and bioconcentration factor (BCF). It should be noted that the BCF is included as a measure of bioaccumulation and is not used to calculate an exposure concentration. Specifically, the measures used are:

- the natural log of the BOD,
- hydrolysis half-lives (in days), and
- and the log of the BCF (unitless).

Although it is recognized that the fate parameters BOD half-life and hydrolysis half-life are not as relevant to chemicals that primarily partition into air, they are used by Swanson et al. (1997) and in this study to represent biotic and abiotic degradation. Furthermore, data for these fate parameters are generally available for a large number of chemicals. U.S. EPA (1994) describes the systematic process in which these parameters are used to develop fate HVs that range from 1 to 2.5.

Some fate measures for the PBT method have been estimated, because experimental data are not found in the databases searched. For example, the BOD and hydrolysis half-lives for  $NO_x$  (as  $NO_2$ ),  $SO_x$  (as  $SO_2$ ), and RDX are estimated to be sufficiently low that they should get the lowest fate HV of 1. These estimated fate HVs are based on fate estimates for similar compounds reported in the chemical-ranking document by the U.S. EPA (1994).

Since fate considerations are considered pivotal to the potential for exposure, the fate HVs are multiplied by the toxicity HV (Swanson et al. 1997). As a result, the final EF for a chemical is based on the formula:

$$EF = (\text{toxicity HV})(\text{BOD HV} + \text{hydrolysis HV} + \text{BCF HV}) \quad (1)$$

Thus, the maximum EF any chemical could have is  $(5)(2.5 + 2.5 + 2.5) = 37.5$ .

Finally, the three types of toxicity EFs are multiplied by associated LCI chemical emissions released to relevant environmental media based on the formula:

$$\text{PBT Score} = (EF)(\text{appropriate LCI emission in pounds}) \quad (2)$$

As such, LCI emissions are matched to EFs as follows:

- the Inhalation Toxicity EF is applied only to air emissions,
- the Aquatic Toxicity EF is applied only to water emissions, and
- the Terrestrial Toxicity EF considers only ingestion exposure.

Calculation of the Terrestrial Toxicity EF has been modified from the approach used by Tolle (1997), to avoid double counting impacts in more than one impact category. Even though terrestrial biota are typically exposed through multiple pathways, only ingestion exposure was evaluated, since it is not reasonable that the same emissions are inhaled and later ingested.

### 2.3 MFM approach

Fate is incorporated into the MFM approach using Mackay Level III models to determine the steady-state concentrations of chemicals in air, water, sediment, soil and biota (Mackay 1991, Mackay, et al. 1992). In contrast to the PBT Approach, the MFM approach allows inventory chemicals to move into media other than the media to which the chemical is released.

For this case study, inventory items are input as chemical emissions assumed to be continuous based on the production of enough RDX for six munitions per day, seven days per week, which is the approximate number required to fill an order for 2,200 munitions in one year. Emissions to three environmental media (air, water, and soil through landfilled materials) were calculated. Based on these emissions, the model estimates the fate of chemicals in five compartments: air, surface water, sediment, soil, and biota. The model requires both unit world parameters and physical-chemical properties to develop these estimates. The unit world area applied is 1 km<sup>2</sup>, including advective and other transport parameters. The assumptions on depth of compartments are: air – 6 km, water – 10 m, soil – 0.2 m, and sediment – 0.03 m.

A set of unit world parameters specific to the state of Tennessee were created to better match state-specific emissions for the case study. The parameters used in this study for the state of Tennessee are:

- average temperature – 12 degrees Celsius,
- surface water area (as a % of the total unit world area) – 2%,
- soil area (as the balance of the unit world area) – 98%,
- rain rate – 1.7E-04 m/hr,
- water content of soil – 28%,
- solids content of soil – 54%, and
- organic carbon content of soil – 2%.

Types of precipitation other than rain (e.g., sleet, snow, etc.) and the type of surface water (e.g., salinity, changes in pH, changes in flow characteristics, effect of freezing and thawing) have not been considered.

In addition to unit world parameters, the MFM algorithms require a set of physical-chemical properties for each inventory item to determine fate in each unit world. The following eight physical-chemical properties (11 data points) were obtained for each chemical assessed: melting point; solubility in water; pKa; vapor pressure; heat of vaporization; octanol-water partition coefficient; Henry's Law Constant; and air, water, soil, and sediment half-lives. Among the properties, the heat of vaporization was used to determine changes in vapor pressure and Henry's Law Constant as a function of temperature.

As indicated below, estimation methods or data for surrogate chemicals were used for some chemicals:

1. *When laboratory values were not available, estimated values were applied.* The U.S. EPA database called Assessment Tools for the Evaluation of Risk (ASTER) proved to be an excellent source of estimated parameters. The solubility of isophorone diisocyanate and the heat of vaporization of petroleum (crude oil) were estimated using the methods listed in Lyman, et al. (1990), Jørgensen, et al. (1998), and Howard and Meylan (1996).

2. *Surrogates were used for two chemicals from the inventory.* Petroleum and Stoddard solvent were assumed to have similar physical chemical properties to 2-methylnaphthalene and benzene, respectively, as recommended in Battelle (1989).

As suggested by Mackay (1991), metals (in this case, aluminum dust and iron) are assumed to have a negligible vapor pressure of 1E-12 Pascals. This assumption did not effect iron, which is not modeled because a value for K<sub>OW</sub> is not available. For aluminum, the equilibrium concentration in air is determined to be negligible. When aluminum is emitted to the environment, the vast majority of the aluminum is determined to reside in the soil at steady-state.

Potential exposure has been assessed for each toxicity impact category, based on the concentrations in air, surface water, soil, sediment, and biota. In the cases of inhalation and aquatic toxicity categories, the steady-state concentrations of chemical in the air and water are used as estimates of the potential exposure to provide a linkage to the toxicity benchmarks, which are also measured in concentration units. For terrestrial toxicity, the potential exposure is estimated as a dose calculated as a sum of exposures anticipated to rodents through inhalation, drinking water, and ingestion of soil as follows:

$$D_i = [\text{Inhalation Dose}] + [\text{Drinking Water Dose}] + [\text{Soil Ingestion Dose}] \quad (3)$$

for chemical *i* such that

$$\text{Inhalation Dose} = \frac{C_{i\text{-air}} R_{\text{inhalation}}}{W} \quad (4)$$

$$\text{Drinking Water Dose} = \frac{C_{i\text{-water}} R_{\text{water ingestion}}}{W} \quad (5)$$

$$\text{Soil Ingestion Dose} = \frac{C_{i\text{-soil}} R_{\text{soil ingestion}}}{W} \quad (6)$$

where<sup>2</sup>:

- $R_{\text{inhalation}}$  = rate of inhalation (assumed to be a deer mouse at 0.025 m<sup>3</sup>/day)
- $R_{\text{water ingestion}}$  = rate of drinking water ingestion (assumed to be a deer mouse at 3.8 grams/day)
- $R_{\text{soil ingestion}}$  = rate of soil ingestion (assumed to be a white footed mouse at 0.088 grams/day)
- $W$  = body weight (assumed to be a deer mouse of 20 grams)

Again, dose has been used as the potential exposure for the terrestrial toxicity category to provide a linkage to the toxicity benchmarks, which are also measured as a dose. Appropriate exposure related doses and concentrations have been used to determine equivalency factors as follows.

The method used to determine multimedia fate model based 'hazard quotients and indices' is adapted from Kolluru

<sup>2</sup> Rates and body weights based on U.S. EPA (1993)

(1996). Kolluru (1996) suggests non-cancer risk may be expressed in terms of a *hazard quotient* (HQ) for a single substance and *hazard index* (HI) for multiple substances and/or exposure pathways. The ratio of exposure to the toxicity benchmark is called a *hazard quotient* (HQ) for dose- or concentration-based benchmarks:

- HQ from Dose-Based Benchmark

$$HQ = \frac{D_i}{BD_i} \quad (7)$$

- HQ from Concentration-Based Benchmark

$$HQ = \frac{C_i}{BC_i} \quad (8)$$

where:

- $D_i$  = dose for the  $i^{\text{th}}$  chemical [mg/(kg-day)]  
 $BD_i$  = benchmark dose for the  $i^{\text{th}}$  chemical [mg/(kg-day)] as in, for example, a lethal dose  
 $C_i$  = concentration in the respective media for the  $i^{\text{th}}$  chemical [mg/m<sup>3</sup>]  
 $BC_i$  = benchmark concentration for the  $i^{\text{th}}$  chemical [mg/m<sup>3</sup>] as in, for example, a lethal concentration

## 2.4 SRA approach

The purpose of the SRA approach was to provide information on the potential for site-specific human health and ecological risks from toxic emissions that could be compared with similar results from the PBT and MFM approaches for LCIA of the three toxicity impact categories described previously. Since there was no intention of conducting a full human health or ecological RA, some of the components in the U.S. EPA's (1998c) final guidelines for ecological RA are discussed below and some were excluded from this SRA.

The first part of a RA is the problem formulation or scoping process that includes obtaining and evaluating information relevant to generation of risk hypotheses to consider (U.S. EPA 1998c). This included the collection of emission sources/quantities and environmental information for HSAAP and the surrounding area. In addition to the LCI document discussed previously (U.S. EPA 1998a), four documents were used that contained detailed information on emission sources, flow rates, and permit requirements for HSAAP (U.S. Army 1978, Woodward-Clyde Federal Services, Inc. 1994, Holston Defense Corporation 1993, Tennessee Water Quality Control Board 1991). Also, the manufacturing process of the RDX-based explosive fill at HSAAP used for the GBU-24 was discussed by Ostic et al. (1995). The LCI data are based on a production run of 2,200 munitions per year, which is approximately 6 munitions per day. Data on water emissions from the single, industrial wastewater treatment plant (IWTP) at HSAAP were obtained from the U.S. EPA's Permit Compliance System (PCS) database. The effluent from the IWTP flows into the Holston River.

Assessment endpoints were selected for evaluation, based on the species of chemicals emitted, the media (air, water, or

land) emissions are released into, the location and density of human populations in the area, and the abundance of terrestrial and aquatic species in the area. Selection of assessment endpoints generally followed guidance in the U.S. EPA's (1998c) final guidelines for ecological RA, which emphasizes ecological relevance, susceptibility to known or potential stressors, and represents management goals. For this study, the management goal criterion was relevance to one of the three toxicity impact assessment categories used for the PBT and MFM approaches. The three primary assessment endpoints selected for the SRA are:

1. Number of humans exposed to air pollutants above their inhalation threshold concentrations.
2. Maintenance of indigenous aquatic biota populations in the Holston River.
3. No increase in human cancer rate due to ingestion of well water from Holston River.

The primary emissions of concern associated with HSAAP were released into the air or the Holston River. Information on solid waste releases was insufficient to model releases from waste disposal sites into the environment. Thus, screening of water emissions against aquatic toxicity benchmarks and air dispersion modeling to determine human population exposure are each discussed below.

## 2.5 Screening of water emissions for SRA

Water emissions at HSAAP associated with production of the intermediate materials or explosive fill for the GBU-24 were screened against benchmarks to determine the potential for toxicity to aquatic biota. Given the large number of compounds emitted at the IWTP, it became necessary to first see if the number of compounds to be modeled could be limited or reduced based on some criteria. The daily loading attributable to GBU-24 production to the Holston River was calculated based on monthly releases for each of the compounds, daily effluent flow rate from the U.S. EPA's National Pollutant Discharge Elimination System (NPDES) discharge monitoring data, and the fraction of total RDX production necessary to fill 2,200 GBU-24 munitions.

Data were obtained from the U.S. Geological Survey on average daily flows in the Holston River for a gauging station near HSAAP over a ten-year period. The diluted contaminant concentration was calculated using the average daily contaminant input information and the Holston River average daily flow. The flow was chosen to be the 7-day, running average, 10<sup>th</sup> percentile flow, which represents poor, but not absolute worst case, conditions. In order to simplify the concentration calculations, it was assumed that (1) the compounds would remain in the aqueous phase and not volatilize or sorb to sediments or suspended matter and (2) the IWTP effluent and the Holston River water would instantly mix.

These diluted contaminant concentrations were then screened against a number of toxicity parameters including freshwater acute and chronic toxicity, or where toxicity information was not available, to Lowest Observed Effect Level (LOEL) concentration data for freshwater organisms. Contaminants that exceeded any of the probable toxic impact

criteria would be subject to a more intensive modeling of fate and effects.

A second set of calculations was undertaken to evaluate the potential carcinogenicity to humans from ingestion of the contaminants in drinking water. For these calculations, it was assumed that the worst case scenario would be ingestion of the contaminated river water in an untreated state. This scenario could involve the indirect use of river water caused by the connection of the river as a source of recharge of a private well drilled adjacent to the river. As a very conservative scenario, it was assumed that no attenuation and no further dilution occur in the aquifer. Using an average daily drinking water ingestion rate (2 L per day) and an average body mass (70 kg) for humans, the effective ingestion rate per unit body mass was calculated. Calculated values were compared to accepted and available carcinogenicity information on: (1) the carcinogenicity risk level for drinking water of one in one million reported in the Integrated Risk Information System (IRIS) and/or (2) the carcinogenic potency  $TD_{50}$  (chronic dose rate producing tumors in half of the test animals) reported in the Carcinogenic Potency Database (CPDB).

## 2.6 Air modeling for SRA

As part of the SRA at HSAAP, air dispersion analyses were performed on major continuous atmospheric point source emissions related to the production of RDX. Point source emissions were identified, characterized, and modeled using the Industrial Source Complex Short Term (ISCST) model. The primary factors motivating the selection of ISCST are: (1) Source term data in the air permits are in a format compatible with ISCST input requirements, (2) The study area is composed of complex terrain with relatively large variation in elevation between receptor locations, and ISCST has the capability of modeling rudimentary effects of variable terrain elevation on the concentration of a vapor cloud plume, and (3) The inhalation toxicity benchmarks for a number of the pollutant species of interest in this study are based on 8-hour averaging periods, and ISCST has the capability to model vapor dispersion with variability in meteorological conditions on an hourly basis.

By selecting ISCST for performing the vapor dispersion calculations, the following assumptions are implicitly inherent to the analysis:

- All emissions are treated as neutrally buoyant.
- All release properties (mass release rate, temperature, species composition) are assumed to be constant. The release rate is simply taken to be equal to the total emission quantity in one year divided by the time in a one-year period.
- Stack downwash and building wake effects have been neglected in this study. ISCST does not include stack downwash effects when the complex terrain option is employed.

Depletion mechanism such as vapor/particulate deposition wet scavenging, and decay due to chemical and physical processes were neglected in this analysis for all chemical spe-

cies. This is a conservative assumption, because it maintains the highest possible level of airborne contaminant for the longest period of time.

Volatile organic compound (VOC) emissions may contain a number of chemical components exhibiting a large variability in inhalation toxicity. This variability makes it difficult to establish toxicity benchmarks for quantifying risk to the human population since aggregate toxicity and synergistic effects between toxic chemicals is not well understood. In an attempt to establish a valid toxicity benchmark for a mixture of VOCs where no one component can be designated as the representative worst-case actor, the NAAQS standard for ozone concentration associated with VOCs was selected as the inhalation toxicity benchmark for VOC mixtures.

The approach taken in this report was to compare the aggregate concentration of a VOC mixture emission to the ozone benchmark with the assumption that all VOCs are transformed to ozone in the time period of interest. Ozone has a much lower inhalation toxicity benchmark than the other constituent chemicals in VOC emissions. Therefore, this approach is conservative in terms of the predicted impact on the human population.

## 3 Results

Data collection and application of each of the three LCIA methods for characterizing toxicity impact potential requires an increasing level of effort beginning with the least effort for the PBT method, intermediate effort for the MFM method, and the greatest effort for the SRA method. Roughly six times more effort is needed for the MFM method than the PBT method. The effort to collect site-specific environmental data and conduct air dispersion modeling on 9 chemicals for the SRA method required about 24 times more effort than the PBT method and about 4 times more effort than the MFM method. These comparative estimates of effort involved the use of experienced researchers for each method and required the collection of physical and chemical data for each of the chemical emissions, plus the collection of environmental characteristics data. The effort for all three methods can be reduced by previously-compiled data on chemicals and the effort for the MFM and SRA methods are further reduced by previously-compiled data on environmental characteristics.

### 3.1 Screening water emissions against benchmarks

The maximum, diluted, water contaminant concentrations estimated in the Holston River as a result of IWTP release of 56 chemicals at HSAAP were screened against acute and chronic toxicity or LOEL benchmarks for freshwater aquatic organisms. Diluted concentrations of toxic chemicals from the IWTP were all well below the benchmarks, so no further modeling was considered necessary for this SRA. The primary caveat is that the calculated concentrations represent only the amount of contaminant attributed to production of the GBU-24, and are not representative of the total concentrations of the contaminant in the river. Actual concentrations would be a function of the background river

concentration and the total emission of any contaminant from all HSAAP discharge points.

A second set of calculations evaluated the potential for carcinogenicity of 25 contaminants in drinking water. For these calculations, it was assumed that the worst case scenario would be ingestion of the contaminated river water in an untreated state by indirect intake of water via recharge from the river into a private well. All of the diluted concentrations in the Holston River were well below the IRIS carcinogenicity risk level of one in one million. Also, the effective human ingestion rate per unit body mass for each contaminant in drinking water was at least seven orders of magnitude below the carcinogenic potency dose rate in test animals. Thus, further modeling was not considered necessary for this SRA.

### 3.2 Air dispersion modeling

Only 2 of the 16 sources for air emissions modeled at HSAAP resulted in offsite ground-level concentrations that exceeded the inhalation toxicity benchmarks. These two sources are the VOC emissions from the building containing the acetic acid distillation process and the combined sulfur dioxide emissions from several boilers co-located in a single building. In addition to these two sources, only one other emission source resulted in a predicted ground-level concentration (onsite or offsite) exceeding the benchmark concentration, namely the acetic acid component of the VOC emissions from a second building. The inhalation toxicity chronic threshold concentrations for VOC and acetic acid, respectively, were selected to be 157 and 250  $\mu\text{g}/\text{m}^3$ . As discussed previously, the inhalation threshold concentration for VOCs is based on the 8-hour NAAQS for ozone. The inhalation threshold concentration for acetic acid is based on an 8-hour, time-weighted average (TWA) of 25  $\text{mg}/\text{m}^3$  divided by an uncertainty factor of 100 for application to the general public.

Concentration isopleths (contours of constant concentration) were developed for VOC air emissions equal to or exceeding the inhalation benchmarks, based on the ISCST receptor concentration results and the averaging period for the inhalation benchmark. The source emission that yielded the benchmark isopleth covering the most offsite area was the VOC release from the acetic acid distillation building, although the benchmark isopleth for VOC emissions from a second building also covered a small area offsite. Concentration isopleths were not developed for the sulfur dioxide release from co-located boilers due to the inherent uncertainties in the analysis and the fact that the release only exceeded the benchmark concentration for one point on the computational grid, which is less than a city block in size.

The concentration isopleth contours developed from ISCST modeling were overlain on top of 1990 population density maps generated for the surrounding area from the U.S. EPA website called Envirofacts. This provided an indication of the population exposed within an isopleth for each pollutant concentration modeled. The isopleths (given in units of  $\mu\text{g}/\text{m}^3$ ) were adjusted to conform to the scale of the population density map and positioned so that the vapor source coincided with the emission source location.

When the population density maps are overlain with the isopleths for the release of total VOCs from the acetic acid distillation building, it suggests that areas of the city of Kingsport near Area A of HSAAP are exposed to ozone concentrations above the 8-hour-average, toxicity benchmark. The benchmark VOC concentration isopleth includes portions of four Kingsport neighborhoods and two schools. Based on the 1990 population density and the area covered by the benchmark VOC concentration isopleth, it is roughly estimated that between 500 to 1,000 people live within the area potentially exposed above the benchmark.

Although much more detailed modeling is required to confirm the assumption that all of the VOCs are converted to ozone in the time period of interest, it is known that the Kingsport area has had episodes when ozone concentrations exceed the NAAQS standards. For example, the AIRS EXEC database, which is maintained by the U.S. EPA, indicated that an air quality monitoring station in Sullivan County exceeded the ozone 1-hour NAAQS two times during the same year as the inventory data. In addition, during the period 1990-1997, the Tennessee Eastman Company, which is located in Kingsport, was the second largest emitter of VOCs in the U.S. Thus, the modeled VOC concentrations from HSAAP are likely to represent only a small portion of the total VOCs in the same area.

The benchmark concentration (250  $\mu\text{g}/\text{m}^3$ ) isopleth generated by air dispersion modeling for release of acetic acid from the second building suggests that the majority of the area above the benchmark is on HSAAP property in Area B. However, according to the modeling results, two neighborhoods in the city of Kingsport are exposed to acetic acid concentrations above the suggested benchmark. Based on the 1990 population density and the area covered by the benchmark concentration isopleth, it is roughly estimated that between 300 to 800 people live within the area potentially exposed above the benchmark.

### 3.3 Comparison of three LCIA approaches

Direct comparison of impact potential scores for the three approaches were limited to inhalation toxicity scores for nine air pollutants, even though there were a total of 21 different types of emissions (18 chemicals and 3 mixtures) to air, water, and/or land (Table 1). As indicated previously, SRA modeling was not conducted for solid wastes or water emissions, respectively, due to insufficient disposal information for modeling or water concentrations below benchmarks after initial dilution. For example, SRA modeling was not conducted on eight of the chemical water emissions, because initial dilution in the Holston River resulted in concentrations that were below the threshold for those chemicals. SRA modeling was not conducted on 9 of the chemicals in the water and 14 of the chemicals on land, because there were no emissions of a specific chemical in that media. No data were provided in the LCI for four chemicals in solid waste or in water, because those four chemicals are air emissions that are not released in the other two media.

**Table 1:** Impact potential scores for emissions from GBU-24 life cycle at HSAAP calculated for three toxicity impact categories by three LCIA characterization methods

Chemical Emissions	PBT Scores			MFM Scores for TN			SRA Scores for HSAAP		
	Inhalation Toxicity	Terrestrial Toxicity	Aquatic Toxicity	Inhalation Toxicity	Terrestrial Toxicity	Aquatic Toxicity	Inhalation Toxicity	Terrestrial Toxicity	Aquatic Toxicity
Acetic Acid (AcOH)	9.6E+00	0.0E+00	3.4E-02	1.2E-10	9.3E-01	1.1E-08	9.2E-01	NE <sup>d</sup>	ECBT <sup>e</sup>
Acetone	2.0E-01	0.0E+00	1.4E-02	4.1E-12	1.6E-05	1.8E-15	2.9E-03	NE	NE
Ammonia	0.0E+00	0.0E+00	2.0E-02	2.9E-20	4.9E-14	4.5E-11	NE	NE	ECBT
CO (Carbon monoxide)	2.6E+01	BM NA	BM NA	5.4E-14	BM NA	BM NA	2.1E-02	NE	NE
Cyclohexanone	4.6E-01	0.0E+00	0.0E+00	1.2E-11	1.3E-06	1.0E-15	2.8E-02	NE	NE
Diethyladipate (DOA 10% of CXM-7)	BM NA <sup>a</sup>	1.4E-02	0.0E+00	BM NA	2.1E-02	3.7E-15	NE	NM	NE
Hydroxide (as Sodium Hydroxide)	BM NA	BM NA	8.2E-03	BM NA	BM NA	7.2E-15	NE	NE	ECBT
Iron	BM NA	BM NA	3.7E-01	BM NA	BM NA	4.0E-15	NE	NE	ECBT
Lead	ND <sup>b</sup>	ND	ND	ND	ND	ND	NE	ND	ND
Methyl Ethyl Ketone (MEK)	ND	ND	ND	ND	ND	ND	NM <sup>c</sup>	ND	ND
Nitric Acid	3.9E-02	0.0E+00	0.0E+00	3.6E-12	4.4E-02	1.7E-10	NM	NE	NE
NO <sub>x</sub> (as Nitrogen Dioxide)	3.7E+02	BM NA	BM NA	2.7E-11	BM NA	BM NA	4.1E-01	NE	NE
Petroleum (Crude Oil)	BM NA	BM NA	2.1E-01	BM NA	BM NA	7.7E-12	NE	NE	ECBT
Phenol	0.0E+00	0.0E+00	8.0E-02	1.4E-23	2.0E-15	1.7E-13	NE	NE	ECBT
PM-10	ND	ND	ND	ND	ND	ND	1.2E-01	ND	ND
Propyl Acetate	BM NA	0.0E+00	0.0E+00	BM NA	3.0E-06	4.6E-13	NM	NE	NE
RDX (Trimethylenetrinitramine or Cyclonite)	BM NA	2.1E+00	0.0E+00	BM NA	2.0E+01	5.0E-08	NE	NM	NE
SO <sub>x</sub> (as Sulfur dioxide)	2.8E+01	BM NA	BM NA	7.5E-10	BM NA	BM NA	1.1E+00	NE	NE
Sulfide (as Sodium Sulfide)	BM NA	0.0E+00	1.0E-01	BM NA	4.8E-18	3.7E-15	NE	NE	ECBT
Sulfuric Acid	0.0E+00	0.0E+00	7.6E+00	6.2E-44	7.7E-13	2.8E-10	NE	NE	ECBT
Volatile Organic Compounds (VOCs)	ND	ND	ND	ND	ND	ND	2.0E+01	ND	ND

<sup>a</sup> BM NA = Benchmark Not Available<sup>b</sup> ND = No Data in LCI<sup>c</sup> NM = Not Modeled due to insufficient data<sup>d</sup> NE = No Emissions in media<sup>e</sup> ECBT = Emission Concentration Below Threshold, so no dispersion modeling conducted**Table 2:** Correlations on rank order of inhalation toxicity potential scores for emissions from GBU-24 life cycle at HSAAP calculated by three LCIA characterization methods

Chemical Emissions	PBT <sup>b</sup>	MFM <sup>c</sup>	SRA <sup>d</sup>
SO <sub>x</sub> (as Sulfur Dioxide)	2	1	1
Acetic Acid (AcOH)	4	2	2
NO <sub>x</sub> (as Nitrogen Dioxide)	1	3	3
Cyclohexanone	5	4	4
CO (Carbon Monoxide)	3	6	5
Acetone	6	5	6
Ammonia	7	7	7
Phenol	7	8	7
Sulfuric Acid	7	9	7

<sup>a</sup> Correlation Coefficient of MFM and SRA Ranks = 0.96;

Correlation Coefficient of PBT and SRA Ranks = 0.83;

Correlation Coefficient of PBT and MFM Ranks = 0.78

<sup>b</sup> PBT = Persistence, Bioaccumulation, and Toxicity Approach<sup>c</sup> MFM = Multimedia Fate Modeling Approach<sup>d</sup> SRA = Simplified Risk Assessment Approach

Correlations were made on the rank order of the impact potential scores for the nine air emissions evaluated for all three LCIA methods (Table 2). Although the number of chemicals compared is very limited, the best correlation coefficient (0.96) was between the rank orders for the MFM and the SRA methods. The correlation coefficient for a comparison of the rank orders for the PBT and SRA methods was 0.83. The lowest correlation coefficient (0.78) was between the rank orders for the PBT and MFM methods. It is assumed that the SRA method using air dispersion modeling is the most accurate of the three methods for predicting potential inhalation toxicity impacts. Based on this assumption, the correlation coefficients for this one case study suggest that the MFM method is more accurate than the PBT method at predicting the same rank order for the impact potential scores as the SRA method. However, the top five chemicals for the PBT method were the same top five chemicals predicted by the SRA method.

#### 4 Conclusions

The PBT, MFM, and SRA methods for characterization of toxicity impact potential were compared using LCI data for

manufacturing of the RDX-based explosive, which is incorporated in the GBU-24 munition. Due to the extensive effort to conduct a site-specific RA, the inventory information evaluated by the three methods was limited to material processing and energy production at HSAAP in Kingsport, Tennessee.

Selection of an appropriate method for LCIA characterization of toxicity impact potential needs to consider the number of sites and emissions involved in the life cycle, as well as the accuracy required for uses of the resulting LCIA information. If the life cycle assessment (LCA) involves numerous sites (e.g., > 20) and numerous chemical emissions (e.g., > 30), the substantial reduction in effort required for the PBT method compared to the other two methods evaluated might justify the potential minor reduction in accuracy of the chemical ranking. Thus, the PBT approach appears to be acceptable as an initial screening approach for evaluating the toxicity impact potential associated with an extremely large LCIA, with little or no previously compiled information on characteristics of chemical emissions or regional/site-specific environmental characteristics. If the accuracy associated with methods for characterizing other impact potentials in the LCIA is somewhat limited, then evaluation of the toxicity impact potentials by a much more accurate and time consuming method, such as the SRA method, may not be justified. On the other hand, if additional accuracy in the toxicity impact potentials are desired, the PBT method could be used to identify the top 10 to 15 chemicals with the highest impact potentials, which could then be evaluated by the MFM method.

As the number of regions/sites and chemical emissions decreases, the improved accuracy of the MFM method is more likely to justify the increase in effort for use of this method instead of the PBT method. Based on the extremely limited number of air emissions (nine chemicals) compared in this study, the correlation coefficient (0.96) for rank order comparison of the MFM method with the SRA method was much better than the correlation coefficient (0.83) for rank order comparison of the PBT and SRA methods. Where chemical and environmental characteristics have been previously compiled for chemicals and regions included in the LCIA, then the improved accuracy of the MFM can be obtained with very little additional effort compared to the less accurate PBT method. The slightly increased effort and improved accuracy of the MFM is more appropriate for LCIA's involving comparative assertions or governmental policy decisions (e.g., involving an entire industry). Unlike the other two methods, the MFM method also has the advantage that it considers transfers to other media, even though our results are too limited to illustrate these intermedia transport reactions. Either the PBT method or the MFM method can be used to identify one or two sites with the most toxicity impact potential, which can subsequently be evaluated with the SRA method using media-specific dispersion modeling.

Although the SRA method is assumed to be the most accurate of the three LCIA toxicity impact characterization methods evaluated, the substantial amount of additional effort for obtaining the environmental characteristics information

needed for modeling does not appear to be justified, unless there are only one or two sites included in the LCIA. This limited number of sites in an LCIA is highly unlikely given the comprehensive nature of most life cycle studies.

In the SRA conducted for this study, only the major air emissions likely to exceed threshold concentrations were modeled. Water emissions were not modeled in this study, since the initial dilution in the Holston River was below threshold concentrations. Solid wastes were also excluded from modeling due to lack of available information required for waste disposal modeling. Site-specific chemical emission data needed for RA modeling is almost never included in LCIs and is difficult to obtain, especially differences in emission rates over time. Thus, aggregation of the impact potentials for aquatic toxicity and terrestrial toxicity into category indicators, which is recommended for all impact categories by ISO (1999), was not possible using the SRA approach in this case study and could be a problem if SRA is used for other case studies.

The SRA in this study (as well as most other types of RAs) compared the modeled environmental concentrations against threshold concentrations (e.g., WQC, NAAQS, RfDs, and MRLs) developed for regulatory and RA purposes. These threshold concentrations are not considered appropriate for an LCIA where individual chemical toxicity scores are combined into a category indicator result (Udo de Haes and Jolliet 1999, Owens 1999, ISO 1999). Threshold concentrations typically include safety factors that are value based and are not strictly science based like an LD<sub>50</sub>. The endpoints are not uniform and may vary from allergic reaction to death. Therefore, even for the air emissions modeled in this study, it is not appropriate to aggregate the results into a category indicator.

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## References

- Battelle Memorial Institute (1989): Multimedia Environmental Pollutant Assessment System (MEPAS) Application Guidance Vol 2: Guidelines for Evaluating MEPAS Input Parameters. Pacific Northwest Laboratory. Richland, WA
- Betts KS (1998): Chemical industry pressured to test high-production volume chemicals. *J Environ Sci Tech* 32, 251A
- Guinée JB, Heijungs R (1993): A proposal for the classification of toxic substances within the framework of life cycle assessment of products. *Chemosphere* 26, 1925-1944

- Hertwich EG, Pease WS, McKone TE (1998): Evaluating toxic impact assessment methods: What works best? *Environ Sci Tech* 3, 138A-144A
- Hogan LM, Beal RT, Hunt RG (1996): Threshold inventory interpretation methodology: A case study of three juice container systems. *Int J LCA* 1, 159-167
- Holston Defense Corporation (1993): Superfund Amendment and Reauthorization Act, Section 313: Toxic Chemical Release Inventory. Prepared for the Holston Army Ammunition Plant to submit to the US EPA by the Holston Defense Corporation, Kingsport, TN
- Howard P, Meylan WM (1996): Prediction of physical properties, transport, and degradation for environmental fate and exposure assessments. Proceedings of the 7<sup>th</sup> International Workshop on QSARs in Environmental Science. Elsinore, Denmark. June 24-28. Downloadable at <http://esc.syrres.com/~ESC/qsar4.htm>
- International Life Sciences Institute (ILSI), Health and Environmental Sciences Institute (1996): Human Health Impact Assessment in Life Cycle Assessment: Analysis by an Expert Panel. Life Cycle Assessment Technical Committee, ILSI, Health and Environmental Sciences Institute, Washington, DC
- International Organization for Standardization (ISO) (1999): Environmental Management – Life Cycle Assessment – Life Cycle Impact Assessment. ISO Standard 14042. Prepared by Technical Committee 207, Sub Committee 5 (ISO/TC 207/SC 5)
- Jørgensen SE, Halling-Soerensen B, Mahler H (1998): Handbook of Estimation Methods in Ecotoxicology and Environmental Chemistry. Lewis Publishers, Boca Raton, FL
- Kimerle RA, Barnhouse LW, Brown RP, de Beyssac BC, Gilbertson M, Monk K, Poremski HJ, Purdy RE, Reinert KH, Rolland RM, Zeeman MG (1997): Ecological effects. pp 89-111. In: Swanson MB, Socha AC. Chemical Ranking and Scoring: Guidelines for Relative Assessments of Chemicals. SETAC Press, Pensacola, FL
- Kolluru RV (1996): Health Risk Assessment: Principles and Practices. Risk Assessment and Management Handbook for Environmental, Health, and Safety Professionals. McGraw-Hill, Inc., New York, NY
- Lyman WJ (1990): Solubility in Water. Handbook of Chemical Property Estimation Methods Environmental Behavior of Organic Compounds. Lyman WJ, Reehl WF, Rosenblatt DH (eds) American Chemical Society. Washington, DC
- Mackay D (1991): Multimedia Environmental Models: The Fugacity Approach. Lewis Publishers, Inc., Chelsea, MI
- Mackay D, Shui WY, Ma KC (1992): Illustrated Handbook of Physical-Chemical Properties and Environmental Fate for Organic Chemicals: Volume I, Monoaromatic Hydrocarbons, Chlorobenzenes, and PCBs. Lewis Publishers, Inc. Chelsea, MI
- Ostic JK, Reardon PT, Parker RY, Gonzales JM (1995): Life-Cycle Inventory for GBU-24 and M-900 Weapon Systems. Draft. Technology Modeling and Analysis Group, TSA-7, Los Alamos National Laboratory, Los Alamos National Laboratory, Los Alamos, NM, pp 72
- Owens JW (1996): LCA impact assessment categories: Technical feasibility and accuracy. *Int. J. LCA* 1, 151-158
- Owens JW (1999): Why life cycle impact assessment is now described as an indicator system. *Int. J. LCA* 4, 81-86
- Society of Environmental Toxicology and Chemistry (SETAC) (1993a): Conceptual Framework for Life Cycle Impact Assessment. Fava JA, Consoli F, Denison RA, Dickson K, Mohin T, Vigon BW (eds) Society of Environmental Toxicology and Chemistry, Pensacola, FL
- Society of Environmental Toxicology and Chemistry (SETAC) (1993b): Guidelines for Life Cycle Assessment: A 'Code of Practice'. Consoli F, Allen D, Boustead J, Fava J, Franklin W, Jensen A, de Oude N, Parrish R, Perriman R, Postlethwaite D, Quay B, Séguin J, Vigon B (eds) Society of Environmental Toxicology and Chemistry, Pensacola, FL
- Society of Environmental Toxicology and Chemistry (SETAC) (1997): Life Cycle Impact Assessment: The State-of-the-Art. Barnhouse L, Fava J, Humphreys K, Hunt R, Laibson L, Noesen S, Owens J, Todd J, Vigon B, Weitz K, Young J (eds). Report of the SETAC Life-Cycle Assessment Workgroup, Society of Environmental Toxicology and Chemistry, Pensacola, FL
- Swanson MB, Davis GA, Kincaid LE, Schultz TW, Bartmess JE, Jones SL, George EL (1997): Screening method for ranking and scoring chemicals by potential human health and environmental impacts. *Environ Toxicol Chem* 16, 372-383
- Tennessee Water Quality Control Board (1991): NPDES Permit No. TN0003671, Holston Army Ammunition Plant, Kingsport, Sullivan County, TN. Tennessee Water Quality Control Board, Tennessee Department of Conservation, Nashville, TN
- Tolle DA (1997): Regional scaling and normalization in LCIA: Development and application of methods. *Int. J. LCA* 2, 197-208
- Udo de Haes HA (Ed) (1996): Towards a Methodology for Life Cycle Impact Assessment. Society of Environmental Toxicology and Chemistry (SETAC)-Europe, Brussels, Belgium
- Udo de Haes HA, Jollier O (1999): How does ISO/DIS 14042 on life cycle impact assessment accommodate current best available practice? *Int. J. LCA* 4, 75-80
- US Army (1978): Environmental Impact Assessment on Production of RDX and HMX Explosives, Holston Army Ammunition Plant. Armament Materiel Readiness Command, U.S. Army, Alexandria, VA
- US Environmental Protection Agency (EPA) (1993): Wildlife Exposure Factors Handbook. EPA/600/R-93/187. Office of Research and Development, Office of Solid Waste and Emergency Response (OSWER), and Office of Water (OW), US EPA, Washington, DC
- US Environmental Protection Agency (EPA) (1994): Chemical Hazard Evaluation for Management Strategies: A Method for Ranking and Scoring Chemicals by Potential Human Health and Environmental Impacts. EPA/600/R-94-177. Risk Reduction Engineering Laboratory, Office of Research and Development, US EPA, Cincinnati, OH
- US Environmental Protection Agency (EPA) (1998a): A Life-Cycle Inventory-Based Comparison of an RDX-Based and TNAZ-Based GBU-24 Munition. Draft Report. Prepared by Battelle on Contract No. CR822956 for the National Risk Management Research Laboratory, Office of Research and Development, US EPA, Cincinnati, OH
- US Environmental Protection Agency (EPA) (1998b): A Life-Cycle Impact Assessment Demonstration for the GBU-24. EPA/600/R-98/070. Prepared by Battelle on Contract No. CR822956 for the National Risk Management Research Laboratory, Office of Research and Development, US EPA, Cincinnati, OH
- US Environmental Protection Agency (EPA) (1998): Guidelines for Ecological Risk Assessment. EPA/630/R-95/002F. Final Risk Assessment Forum, US EPA, Washington, DC
- Woodward-Clyde Federal Services, Inc. (1994): Final Air Pollution Emissions Statement, Holston Army Ammunition Plant, Kingston, Tennessee. Submitted to the US Army Environmental Center by Woodward-Clyde Federal Services, Inc., Rockville, MD